Denitrification from a Swine Lagoon Overland Flow Treatment System at a Pasture–Riparian Zone Interface

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ABSTRACT

In manure disposal systems, denitrification is a major pathway for N loss and to reduce N transport to surface and ground water. We measured denitrification and the changes in soil N pools in a liquid manure disposal system at the interface of a pasture and a riparian forest. Liquid swine manure was applied weekly at two rates (approximately 800 and 1600 kg N ha^{-1} yr^{-1}) to triplicate plots of overland flow treatment systems with three different vegetation treatments. Denitrification (acetylene block technique on intact cores) and soil N pools were determined bimonthly for 3 yr. The higher rate of manure application had higher denitrification rates and higher soil nitrate. Depth 1 soil (0–6 cm) had higher denitrification, nitrate, and ammonium than depth 2 soil (6–12 cm). The vegetation treatment consisting of 20 m of grass and 10 m of forest had lower denitrification. Denitrification did not vary significantly with position in the plot (7, 14, 21, and 28 m downslope), but nitrate decreased in the downslope direction while ammonium increased downslope. Denitrification ranged from 4 to 12% of total N applied in the manure. Denitrification rates were similar to those from a nearby dairy manure irrigation site, but were generally a lower percent of N applied, especially at the high swine effluent rate. Denitrification rates for these soils range from 40 to 200 kg N ha^{-1} yr^{-1} for the top 12 cm of soil treated with typical liquid manure that is high in ammonium and low in nitrate.

Confinement production operations for swine, dairy, and poultry generate large amounts of manure in many parts of the USA and other developed countries. Although land application for crop production is the preferred method of managing the manure, this is not always possible. In some cases the goal of manure management will be treatment of the manure with crop production goals being either secondary or nonexistent. The most common example of on-farm treatment of manure has been the use of constructed wetlands for removal of nutrients and organic C from liquid manure. Although constructed wetlands offer the possibility of high levels of manure effluent treatment, they have a number of disadvantages including (i) expense, (ii) difficulty of proper operation, and (iii) potential for nutrient accumulation. In addition, constructed wetlands offer only limited opportunities for crop production with the manure resource. Overland flow treatment systems have been proposed as a possible alternative to constructed wetlands where there is not an adequate land base for manure use for crop production and when constructed wetlands are not acceptable to the farmer. Although regulatory and acceptability hurdles for overland flow treatment systems are recognized, more information is needed on the performance, water quality impacts, and sustainability before they can be implemented as a possible water quality best management practice (BMP). This study and the related studies (Hubbard et al., 1998) are designed to test overland flow treatment as a manure disposal and treatment alternative.

Although it is the preferred method of handling manure, land application of manures has been associated with (i) water quality degradation, (ii) air pollution through N gas emission, and (iii) other problems such as odors and potential movement of pathogenic organisms (Edwards et al., 1996; Hubbard et al., 1987; Wallingford et al., 1975; Westerman and Overcash, 1980).

One of the most common practices for swine and dairy production is to use gravity fed water to flush manure from confinement facilities (Merkel, 1981; Sweeten and Wolfe, 1994). The flush water is routed to one or a series of storage lagoons. Facilities of this sort have been in common use for the past 30 yr and are eligible for cost-share funds and technical assistance in many watershed and water quality improvement projects sponsored by USDA and other agencies.

While providing efficient and cost effective manure collection, flush facilities and lagoons produce a relatively large volume of dilute liquid manure from a relatively small mass of concentrated feces and urine. Proper management of the liquid manure involves application of the nutrients needed for crop growth, nutrient removal in a treatment system, or a combination. In addition to uptake and removal in crop harvest, N is lost from manure management systems in gaseous forms. This generally takes two forms—the volatilization of ammoniacal N as ammonia gas and the loss of N gases through denitrification. Denitrification is the microbial respiration of nitrate and the evolution of dinitrogen (N_2), nitrous oxide (N_2O), or nitric oxide (NO) as the by-product of this respiration (Tiedje, 1982). Denitrification is stimulated by liquid manure and manure slurry additions because of the addition of N, C, and water and because of the increased anaerobiosis created (Christensen, 1985; Thompson, 1989).

In this study, we quantified the amount of microbial denitrification occurring in an overland flow treatment system for liquid swine manure. The liquid manure was applied at two rates to three different vegetation treatments. Concurrently, we quantified a number of factors that are critical to control of denitrification from the soils where liquid manure was applied. Among these factors were the nitrate, ammonium, and water contents of the soil. The results are presented and discussed relative to soil depth and position in the plots downslope

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Abbreviations: BMP, best management practice; WFPS, water filled pore space.

from the manure application sites. Based on this and related studies, general conclusions are made concerning the maximum amount of denitrification that is likely to occur from these lagoon effluent application systems.

MATERIALS AND METHODS

Field Sites and Treatments

The investigation was carried out using swine lagoon wastewater from the waste treatment–storage system of the University of Georgia Coastal Plain Experiment Station main swine unit at Tifton, GA. The lagoon system consisted of three lagoons, in series. Liquid manure was pumped from the end of the third lagoon opposite the infall to the site where the plot studies were conducted. Further details of the swine facilities and the lagoon system are found in Hubbard et al. (1998).

The plot studies were conducted at an established grass buffer–riparian forest site approximately 760 m from the swine buildings. The site included a grassed area that had formerly been the lowest end of a pasture for beef cattle, and an adjacent downslope riparian forest with slash pine trees (*Pinus elliottii* Engelm.) and accompanying underlying shrubby vegetation. The soil of the grassed area was Tifton loamy sand (fine-loamy, siliceous, thermic Plinthic Kandiudult) while that of the riparian forest area was an Alapaha loamy sand (loamy, siliceous, thermic Arenic Plinthic Paleaquult) or an intergrade between it and Tifton loamy sand. At a 1- to 2-m depth, these soils are underlain with plinthite and Miocene age materials of very low permeability. Past research has shown that in the plinthic soils of the Tifton Upland, 99% of infiltrating water moves downslope as shallow lateral flow (Hubbard and Sheridan, 1983). The slope at the site ranged from 1.5 to 2.0%.

Three combinations of vegetation at two wastewater application rates were used for the study in 4 m wide by 30 m long plots. The three vegetative treatments were (i) 10 m grass buffer draining into 20 m existing riparian zone vegetation, (ii) 20 m grass buffer draining into 10 m existing riparian zone vegetation, and (iii) 10 m grass buffer draining into 20 m maidencane (*Panicum hemitomon* Schultes) (Fig. 1). Different combinations of grass lengths and riparian forest vegetation were used to determine the relative importance of grass vs. forest in N assimilation. Maidencane is a species recommended for constructed wetlands. The purpose of using the maidencane was to see if wetland plant species other than trees and understory vegetation would be effective in assimilating N.

Coastal bermudagrass (*Cynodon dactylon* L. Pers.) ('Tifton 78') was used for the grassed portion of the plots. During the fall of 1993, 'Georgia 5' fescue, a heat-tolerant tall fescue (*Festuca arundinacea* Schreb.) was planted as a perennial winter cover in the grassed portion of the plots. During the winter of 1995–1996 crimson clover (*Trifolium incarnatum* L.) was seeded on the plots, since the fescue had not performed well in terms of cover during the previous winter (1994–1995). The forested part of the plots had slash pine trees that were 4 yr old and about 2 m tall at the start of the study in Oct. 1993, and approximately 10 m tall by the end of the study in 1996. Maidencane for the study was planted as rhizomes during the summer of 1993. Three or four cuttings of the grassed zone were made each summer and the biomass was completely removed from the plots. Two cuttings (at approximately 30 cm height) were made of the maidencane during the summers of 1995 and 1996 and the biomass was removed. There were three replicates of each vegetative treatment at each wastewater application rate for a total of 18 plots (Fig. 1).

The two wastewater application rates used for the study were either a single application weekly (1×) or two applications per week (2×). Each application consisted of 1285 L plot^{-1}. This corresponded to 1.07 cm of wastewater per plot at each application. Analyses of wastewater samples collected weekly during the study showed that the average N concentration was 160 mg L^{-1}, with most of the N in the NH_4–N form. Nitrate concentrations in the wastewater ranged from <1 to 20 mg N L^{-1} with a mean of 3 mg N L^{-1}. Total N application for the denitrification sampling period was 4335 kg N ha^{-1} at the 2× wastewater application rate and 2176 kg N ha^{-1} at the 1× wastewater application rate. On an annual basis, the 1× plots received 800 kg N ha^{-1} yr^{-1} while the 2× plots received 1600 kg N ha^{-1} yr^{-1}.

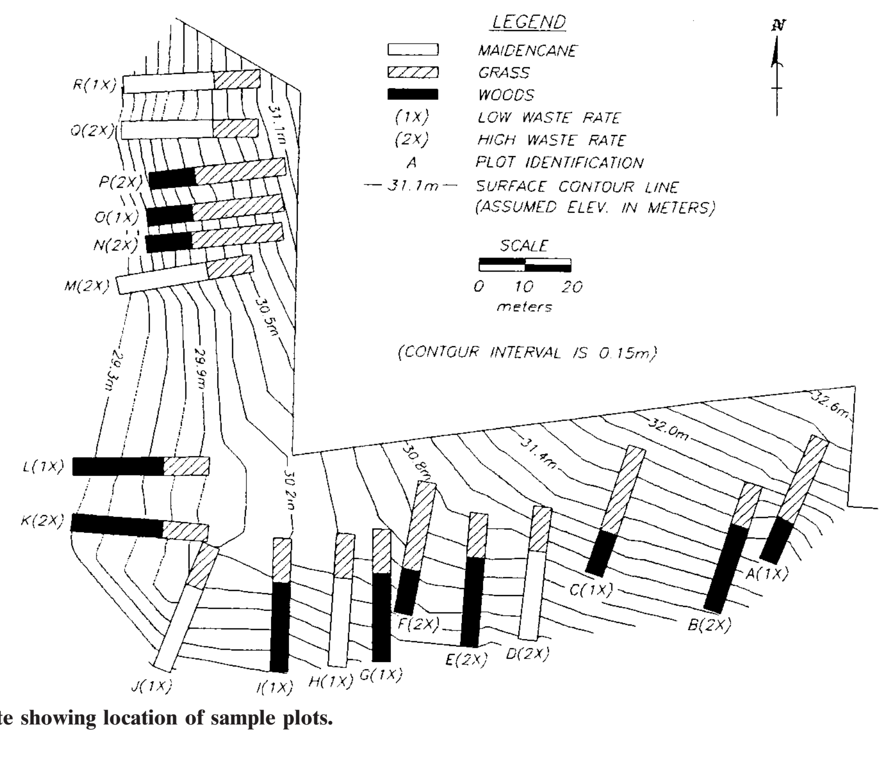

Fig. 1. Map of study site showing location of sample plots.

The overland flow–riparian buffer plots were positioned on the landscape according to contour, so that flow of the wastewater downslope would be as uniform as possible. The sides of each plot were bounded with plastic borders that extended 15 cm aboveground and 15 cm belowground. The purpose of these borders was to prevent overland flow from leaving the plots during wastewater application, or surface runoff from entering from outside of the plots during rainfall/runoff events. An earthen berm at the top of each plot prevented rainfall-induced surface runoff from entering from upslope of the plot. At the top end of each plot a gated pipe made of plastic downspout was used to apply the wastewater. Wastewater flowed from an individual tank above each plot into the gated pipe and then downslope. The pipe gates were spaced 46 cm apart and had adjustable openings. These were adjusted on each plot so that overland flow would begin downslope movement as uniformly as possible. Depending on soil moisture and vegetative conditions, wastewater flowed over one-half to two-thirds of each plot during application. Wastewater was not applied during rainstorms or if it appeared that rainfall was imminent. During the wet winter months, when the soil was nearly saturated, the wastewater was applied very slowly, to minimize any potential for the waste to directly exit the plots. At other times the wastewater was applied as quickly as the tanks would drain completely, resulting in application times of approximately 10 min. The individual plots were spread out on the landscape as much as possible to minimize cross contamination of the shallow ground water from different wastewater application rates and vegetative treatments (Fig. 1).

Soil Sampling and Analysis

Soil samples were taken for measurement of denitrification, soil inorganic N, and soil water content using established procedures (Lowrance and Smittle, 1988; Lowrance, 1992; Lowrance et al., 1995, 1998). Samples were taken once before manure application began (July 1993) and bimonthly for 3 yr after manure application began (November 1993–September 1996). This sampling regime provided samples at 19 different points in time over the 3-yr period. Including the July 1993 sampling (before application began), a total of 2880 soil cores were collected for the study. Soil samples were taken at random with respect to manure application, except to avoid taking soil samples in the middle of a manure application period. On each sample date, intact soil cores (2.5 cm diam.) were taken at four positions in each plot. The positions, numbered 1–4, were 7, 14, 21, and 28 m from the top of each plot. The intact soil cores were taken at 6-cm increments to 12 cm and placed in 60-mL plastic syringes for incubation. The incubation syringes were returned to the laboratory immediately after sampling and the head space on each sample was adjusted to 30 mL. A serum stopper was placed on the tip of the incubation syringe and 3 mL of acetylene was added to the head space after removal of 3 mL of air. Acetylene blocks the reduction of N_2O to N_2 gas as the final reduction step in the denitrification process (Tiedje, 1982). Although the acetylene block technique may underestimate denitrification due to either inhibition of nitrification (Nielsen et al., 1996) or scavenging of NO by acetylene (Bollman and Conrad, 1997), it provides an easily applied technique for multiple estimates of denitrification. The cores were incubated at 25°C for 5 h with 5-mL head space gas samples taken at 1 and 5 h. All incubations were done on the same day the cores were taken. Incubations were done at 25°C to approximate the mean annual soil temperature in the top 30 cm of soil. Soil temperatures were monitored at a weather station 0.5 km from the research site. Values of Q_{10} for denitrification vary widely. Denitrification values were corrected for actual soil temperature based on a Q_{10} of 2, which is near the mean of a number of studies (Ambus, 1993; Peterjohn, 1991). A Q_{10} of 1.6 computed for nitrous oxide emission for a sandy loam soil has been reported, further indicating that a Q_{10} of 2 is appropriate (Smith et al., 1998).

Analysis for N_2O concentration in the head space was done using the electron capture detector of a Varian 3600 gas chromatograph. The gas chromatograph was operated using a 5-m Poropak Q column with a detector temperature of 350°C, oven and injector temperatures of 70°C, and Ar/CH_4 (95%:5%) as a carrier at a rate of 20 mL min^{-1}. All results were corrected for the solubility of N_2O in water based on values given in Moraghan and Buresh (1977). Time 2 (5 h) samples were corrected for the 5 mL removed from the head space in the Time 1 (1 h) sample.

After incubation, the length (L) and mass (G) of each soil core was determined. A 12.00 g subsample was then analyzed for NH_4^+–N and (NO_2^- + NO_3^-)–N following extraction, with 40 mL of a 2 M KCl solution (Keeney and Nelson, 1982). Extracts were analyzed colorimetrically for NH_4^+–N and NO_3^-–N using Lachat Methods 12-107-06-2-A and 12-107-04-1-B, respectively. Gravimetric soil water content (θ_m) was determined by drying the remaining field moist soil at 105°C for 3 d and reweighing. Bulk density (P_B) was determined for each soil core based on the total mass of dried soil (including the portion removed for KCl extraction) and the volume of the core ($V = \pi r^2 L$). Total porosity (TP) was determined as

$$\mathrm{TP} = (1 - P_B/P_p)$$

where P_p = particle density, assumed to be 2.65 Mg m^{-3}. Percent water filled pore space (WFPS) was calculated as

$$\mathrm{WFPS} = [\theta_m G/(\mathrm{TP}\ V)] \times 100$$

Each 6-cm core sample was used to calculate a daily rate (g N_2O–N ha^{-1} d^{-1}). Total denitrification loss in the 12-cm profile was the sum of the g N_2O–N ha^{-1} d^{-1} rates for the two cores of the profile. This calculation assumes that all gases produced in the top 12 cm would eventually leave the soil surface and be lost to the atmosphere. These daily rates for each 12-cm sample were multiplied by the time period between samples (typically around 60 d) and then summed for the entire sampling period (3 yr) to estimate a denitrification rate for the entire sampling period. Average annual rates were calculated by dividing the total denitrification by 3 yr. Annual denitrification rates were compared with annual rates from a nearby liquid dairy manure application site (Lowrance et al., 1998).

Denitrification, soil inorganic N, soil water content, and water-filled pore space data were not normally distributed and were analyzed using nonparametric techniques as was done in similar studies (Lowrance et al., 1995, 1998). Data were analyzed using the NPAR1WAY Procedure of SAS (SAS Inst., 1989) using the Kruskal-Wallis test. NPAR1WAY is a nonparametric procedure for testing whether the distribution of a variable has the same location parameter across different groups. The Kruskal-Wallis procedure tests the null hypothesis that the groups are not different from each other by testing whether the rank sums are significantly different based on a chi-square distribution (Sokal and Rohlf, 1981). In testing for differences among vegetation types, position, depth, or N application rates using the NPAR1WAY procedure, the denitrification rate, soil inorganic N, or WFPS for each core was used as an individual observation. Correlation analysis was done for denitrification vs. soil variables using the correlation procedure of SAS (SAS Inst., 1989).

Table 1. Denitrification, soil nitrate, soil ammonium, and water-filled pore space for Depths 1 and 2 in the swine manure application plots with plots, sample dates, and positions pooled.

Depth	Denitrification	Soil nitrate	Soil ammonium	Water-filled pore space
	g ha^{-1} d^{-1}	mg NO_3–N kg^{-1}	mg NH_4–N kg^{-1}	%
0–6 cm	135 (12.3, 1356)†	7.10 (0.38, 1355)†	18.7 (0.55, 1352)†	89.9 (0.47, 1354)†
6–12 cm	70.2 (6.17, 1358)	4.51 (0.23, 1358)	12.2 (0.40, 1357)	87.3 (0.46, 1354)

† Different between depths at the 0.001 level based on the Kruskal-Wallis test. Values are means followed by standard error and number of observations.

RESULTS AND DISCUSSION

Denitrification, Soil Inorganic Nitrogen, and Soil Moisture

Denitrification rate, nitrate, and ammonium concentrations were all higher in the 0- to 6-cm soil layer than in the 6- to 12-cm soil layer (Table 1). Water filled pore space was higher in depth 2. The manure infiltrated after traversing one-half to two-thirds of the plot on most application dates so that there was substantial manure entering through infiltration over the upper part of the plot. Especially under wetter conditions, the infiltrated liquid manure would tend to stay in the surface soil. In addition, solids that would be codeposited during infiltration would also stay in the surface few centimeters of the soil. Previous studies have shown that about 60% of the denitrification in a liquid manure application system was in the top 12 cm of a 30-cm soil profile (Lowrance et al., 1998), so the majority of the denitrification occurring in this system was probably measured.

Denitrification and soil nitrate were significantly different for the two rates of liquid manure application (Table 2). Soil ammonium and WFPS were not different among the two N application rates. Although a mass balance is beyond the scope of this study, it appears that a combination of increased denitrification and increased nitrate content of soil is diminishing the soil ammonium differences between the treatments. Both the 1× and 2× treatments received large water subsidies (42 and 84% of mean annual rainfall, respectively), so both treatments had consistently high WFPS.

Denitrification and soil ammonium were significantly different among the vegetation treatments (Table 3). Denitrification in the two treatments with 10 m of grass and 20 m of either forest or maidencane were higher than in the treatment with 20 m of grass and 10 m of forest (Table 3). Soil nitrate was not different among the treatments but soil ammonium was lower for the 20 m grass–10 m forest treatment. Because the plots with 20 m grass tended to start at a higher landscape position than the plots with 10 m of grass (Fig. 1), it is difficult to determine the effects of vegetation on a process such as denitrification, which is highly dependent on soil moisture status. Although WFPS was not different among vegetation types, the gravimetric soil moisture was lower in the vegetation treatment of 20 m of grass and 10 m of forest.

Denitrification was not different among the four sampling positions within the plots (Table 4). All positions had similar denitrification rates, which were comparable to rates measured in a nearby liquid dairy manure land application system (Lowrance et al., 1998). Soil nitrate and soil ammonium showed opposite patterns with soil nitrate decreasing significantly from 7 to 28 m below the top of plot and soil ammonium increasing significantly (Table 4). Nitrate decreased consistently from the top of the plot to the bottom, whereas ammonium increased consistently. The different patterns probably reflect both the different transport mechanisms for the two N forms and the potential for high water contents to increase denitrification while decreasing nitrification. Although there was attenuation of both nitrate and ammonium concentrations in surface runoff between upslope and downslope positions, there were no significant reductions in surface runoff total N (Hubbard et al., 1998). It is likely that increases of ammonium downslope are due to a combination of total N transport to the lower portions of the plot and low rates of nitrification in the lower portions of the plot where the soils are wetter. The positions nearest the effluent application (7 m from

Table 2. Denitrification, soil nitrate, soil ammonium, and water-filled pore space for the two swine manure application rates with depths, sample dates, and positions pooled.

Nitrogen rate	Denitrification	Soil nitrate	Soil ammonium	Water-filled pore space
kg N ha^{-1}	g ha^{-1} d^{-1}	mg NO_3–N kg^{-1}	mg NH_4–N kg^{-1}	%
1× (2176)	83.4 (7.16, 1358)†	4.23 (0.22, 1358)†	16.0 (0.51, 1354)ns‡	88.2 (0.48, 1354)ns
2× (4335)	122 (11.8, 1356)	7.35 (0.39, 1355)	14.8 (0.46, 1355)	89.0 (0.46, 1354)

† Different between rates at the 0.001 level based on the Kruskal-Wallis test.
‡ No significant difference between rates. Values are means followed by standard error and number of observations.

Table 3. Denitrification, soil nitrate, soil ammonium, and water-filled pore space for the three vegetation types in the swine manure application plots with rates, sample dates, and positions pooled.

Vegetation	Denitrification	Soil nitrate	Soil ammonium	Water-filled pore space
	g ha^{-1} d^{-1}	mg NO_3–N kg^{-1}	mg NH_4–N kg^{-1}	%
10 m grass, 20 m forest	121 (13.8, 909)†	5.34 (0.45, 909)ns‡	15.2 (0.62, 907)†	89.1 (0.60, 906)ns
10 m grass, 20 m cane	119 (13.6, 895)	6.00 (0.33, 894)	18.5 (0.71, 894)	87.9 (0.53, 893)
20 m grass, 10 m forest	67.9 (7.6, 910)	6.03 (0.38, 910)	12.6 (0.41, 908)	88.9 (0.56, 909)

† Different among vegetation types at the 0.001 level based on the Kruskal-Wallis test.
‡ No significant difference among vegetation types. Values are means followed by standard error and number of observations.

Table 4. Denitrification, soil nitrate, soil ammonium, and water-filled pore space for positions 1 to 4 in the swine manure application plots with plots, sample dates, and depths pooled.

Position	Denitrification	Soil nitrate	Soil ammonium	Water-filled pore space
m from top	g ha^{-1} d^{-1}	mg NO_3–N kg^{-1}	mg NH_4–N kg^{-1}	%
7	108 (15.8, 680)ns‡	11.2 (0.58, 680)†	12.0 (0.58, 678)†	90.5 (0.57, 679)†
14	104 (14.6, 678)	6.29 (0.57, 678)	14.9 (0.56, 676)	86.2 (0.64, 675)
21	103 (13.6, 679)	2.85 (0.19, 678)	15.5 (0.61, 678)	88.2 (0.68, 678)
28	96.5 (10.9, 677)	2.78 (0.19, 677)	19.2 (0.92, 677)	89.7 (0.70, 676)

† Different among positions at the 0.001 level based on the Kruskal-Wallis test.
‡ No significant difference among positions. Values are means followed by standard error and number of observations.

top) and nearest the bottom of the plot had highest water-filled pore space.

Denitrification was significantly correlated with WFPS at the 0.0001 level with Pearson correlation coefficients ranging from 0.21 to 0.29 for the entire data set and for subsets by depth, rate, vegetation, and position. Although the WFPS was generally >60% for these soils, there was still a significant relationship between denitrification and the volumetric water content as has been observed in other studies (Lowrance et al., 1998). There was no correlation between denitrification and soil nitrate. The correlation between denitrification and soil ammonium 7 m from the top of plots was significant at the 0.001 level with a Pearson correlation coefficient of 0.12. Other studies of denitrification and manure application have shown higher correlations with soil ammonium and generally no correlation with soil nitrate (Loro et al., 1997; Lowrance et al., 1998).

Nitrogen Removal by Denitrification

Annual denitrification rates for the 18 plots had a wide range with rates differing by nearly a factor of 5 from 38 to 195 kg N ha^{-1} yr^{-1} (Fig. 2). Although plots with the two highest annual rates were under the 2× treatment (Plots B & D), the third highest rate for a plot was a 1× treatment (Plot J). The average annual denitrification rates for the 1× and 2× treatments ranged from 46 to 125 kg N ha^{-1} yr^{-1} (Fig. 3). Denitrification from the 2× rate was generally higher than from the 1× rate, although the difference was only significant for the vegetation system of 10 m grass/20 m forest. Denitrification rates were generally much higher than in typical coastal plain soils receiving inorganic fertilizer (Lowrance and Smittle, 1988; Lowrance, 1992). The plot rates and mean rates were similar to mean annual rates for the nearby dairy manure application system that had rates ranging from 44 to 187 kg N ha^{-1} yr^{-1} in the top 12 cm of soil. The highest mean annual rate measured from the dairy was higher than any of the mean rates for this study (Fig. 3), but all mean annual rates for the dairy study were in the range of the individual plots receiving swine effluent (Fig. 2). It should be noted that the application rates for the dairy effluent study were less than in this study of swine effluent, and the application method was different.

Denitrification as a percent of total N applied ranged from 4 to 12% (Fig. 4). The highest percent removal was for the 1× rate. For the three vegetation treatments, percent removal was higher for the 1× case in two treatments and for the 2× case in the other treatment. These percent of N removal were generally less than the percent N removed in the liquid dairy manure appli-

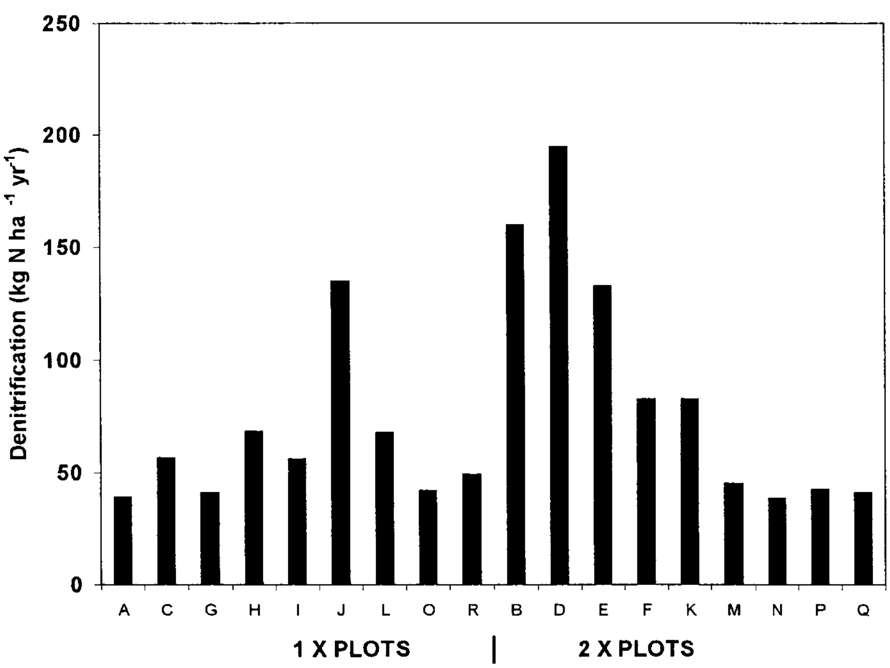

Fig. 2. Mean annual denitrification for each of the 18 plots.

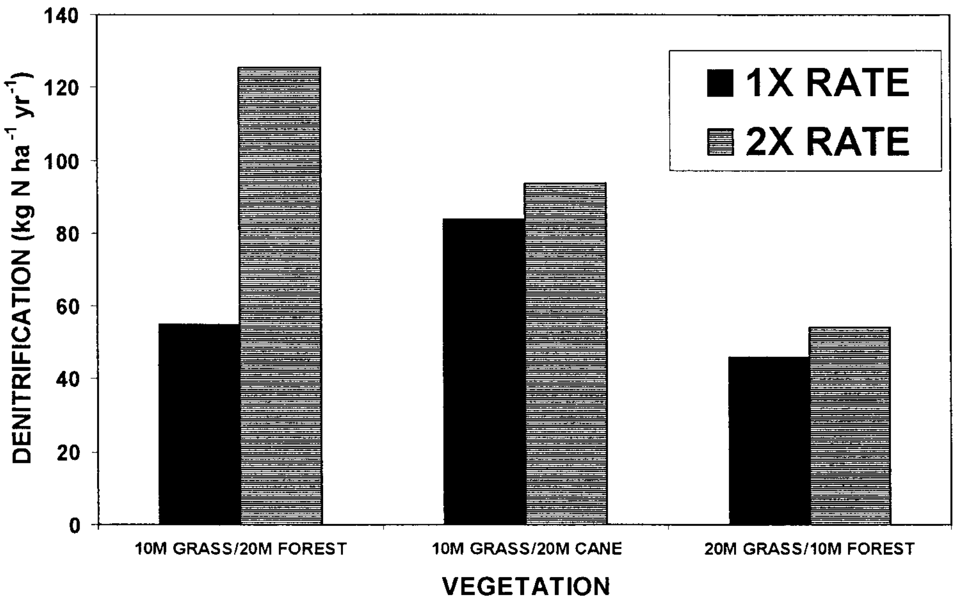

Fig. 3. Mean annual denitrification for each vegetation treatment and N treatment rate.

cation system that had percent removals that ranged from 10 to 29% in the top 12 cm of soil (Lowrance et al., 1998). There were a number of differences in the two studies that would lead to differences in the percent of N removed. In the dairy effluent study, the manure was applied uniformly over the plot and was sometimes supplemented with fresh water. One quadrant of the dairy effluent study (receiving 643 kg N ha^{-1} yr^{-1}) had much higher soil moisture than the other areas. These high soil moisture levels stimulated denitrification (Lowrance et al., 1998).

Very few estimates of denitrification have been made for manure management systems. Denitrification measured in sieved soil placed in large wooden containers accounted for 1.0 to 1.3% of the total N added from liquid dairy manure, which was injected into the soil (Comfort et al., 1990). Nitrous oxide flux accounted for about 1% of the manure (animal excrement and straw bedding) N applied to a maize (*Zea mays* L.) crop in Ontario (Lessard et al., 1996). In contrast with these two studies showing very low denitrification, Thompson et al. (1987) found that total denitrification losses from soil injected with cattle slurry were as high as 46 kg N ha^{-1} accounting for 7 to 21% of the N applied. Although a manure application system in British Columbia had most NO_3^- loss by leaching, up to 48 kg N ha^{-1} was lost via denitrification in 6 mo, accounting for up to 17% of the N applied (Paul and Zebarth, 1997a). As much as 156 kg N ha^{-1} was lost via denitrification during the growing season (Paul and Zebarth, 1997b). Limited information on denitrification in manure management systems indicates that slurries and liquid manure would lead to the highest denitrification rates.

Incubations using the acetylene inhibition technique

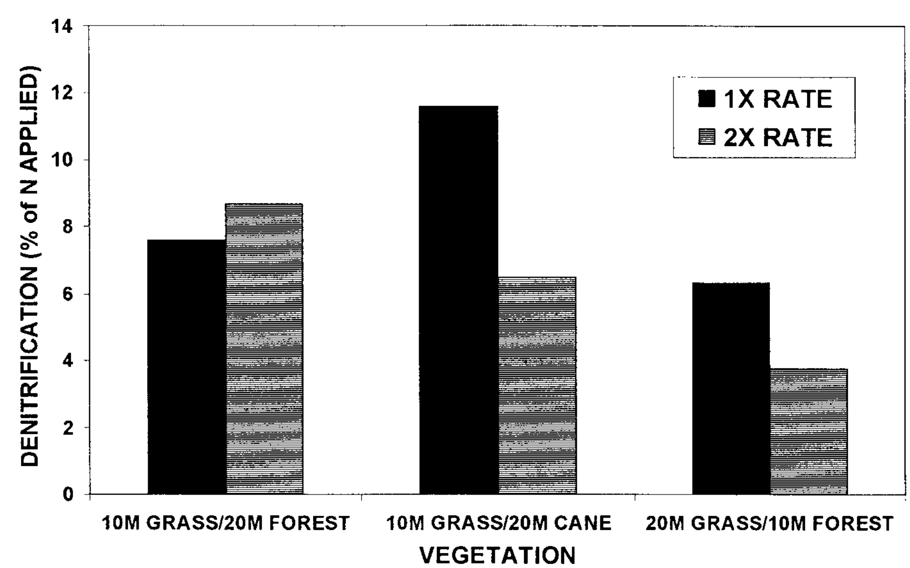

Fig. 4. Denitrification as a percent of N applied for the entire study period.

might underestimate denitrification in manured soils due to elimination of coupled nitrification/denitrification (Nielsen et al., 1996). Conversely, other studies have shown that nitrification did not contribute significantly to nitrate flux to manure hot-spots and that inhibiting nitrification would not affect denitrification (Petersen et al., 1991). Elimination of coupled nitrification/denitrification does not appear to have limited denitrification in this liquid manure application system. Short-term incubation, as used in this study, will have less tendency to limit denitrification due to inhibition of nitrification. The estimated annual denitrification rates are considerably higher than estimates on similar soils with similar soil nitrate levels where nitrate fertilizer and fresh water irrigation were applied (Lowrance and Smittle, 1988; Lowrance, 1992). Nitrate levels in these soils were much higher than in soils of a nearby wetland (Lowrance et al., 1995) but were similar to levels in the dairy effluent irrigation site. Although 2.5-cm cores are small compared with plot size, they encompass most of the important processes controlling denitrification and allow diffusion of acetylene into the soil volume. Small cores also allow processing of large numbers of cores to sample the spatial and temporal variability of denitrification and related soil properties without causing major changes in the systems being studied.

SUMMARY AND CONCLUSIONS

Denitrification was a significant but variable output from the overland flow treatment system. Denitrification losses were as high as 195 kg N ha^{-1} yr^{-1} from the top 12 cm of soil. When compared with denitrification measured using the same methods in a nearby liquid dairy manure land application site, the annual rates were similar but the percent of N loss due to denitrification was lower, because the N applied was greater in the swine plots. Data from both of these studies indicate that about 200 kg N ha^{-1} yr^{-1} may be the maximum denitrification rate possible in the liquid manure application systems on Coastal Plain soils where most of the N in manure is ammonium. Until other data are available, we recommend using 200 kg N ha^{-1} yr^{-1} as the maximum N loss from a liquid manure application system on these soils, even at very high application rates such as the 1600 kg N ha^{-1} yr^{-1} used in this study.

The overland flow land application system had generally good water quality relative to nitrate losses (Hubbard et al., 1998). A consistent pattern of decreasing soil nitrate observed in the plots moving from upslope to downslope supported this conclusion. Conversely, there was a consistent build-up of soil ammonium from the top of the plot to the bottom. Either direct transport of the ammonium or eventual transport as nitrate may contribute to long-term water quality degradation.

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Correlation of Human Olfactory Responses to Airborne Concentrations of Malodorous Volatile Organic Compounds Emitted from Swine Effluent

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ABSTRACT

Direct multicomponent analysis of malodorous volatile organic compounds (VOCs) present in ambient air samples from 29 swine (*Sus scrofa*) production facilities was used to develop a 19-component artificial swine odor solution that simulated olfactory properties of swine effluent. Analyses employing either a human panel consisting of 14 subjects or gas chromatography were performed on the air stream from an emission chamber to assess human olfactory responses or odorant concentration, respectively. Analysis of the olfactory responses using Fisher's LSD statistics showed that the subjects were sensitive to changes in air concentration of the VOC standard across dilutions differing by approximately 16%. The effect of chemical synergisms and antagonisms on human olfactory response magnitudes was assessed by altering the individual concentration of nine compounds in artificial swine odor over a twofold concentration range while maintaining the other 18 components at a constant concentration. A synergistic olfactory response was observed when the air concentration of acetic acid was increased relative to the concentration of other VOC odorants in the standard. An antagonistic olfactory response was observed when the air concentration of 4-ethyl phenol was increased relative to the other VOC odorants in the standard. The collective odorant responses for nine major VOCs associated with swine odor were used to develop an olfactory prediction model to estimate human odor response magnitudes to swine manure odorants through measured air concentrations of indicator VOCs. The results of this study show that direct multicomponent analysis of VOCs emitted from swine effluent can be applied toward estimating perceived odor intensity.

Modern swine management practices have undergone extensive changes during the last two decades in an effort to improve animal production efficiency, reduce animal mortality, and provide safer, higher quality animal products (Barker et al., 1996). These improvements in production efficiency have transformed the infrastructure of the swine industry, and have permitted the effective management of larger populations of animals on production sites. The expansion of concentrated animal feeding operations (CAFOs) throughout the USA has catalyzed an increased awareness by the general public and governmental agencies for the potential effects of these facilities on water and air quality (Schiffman et al., 1995; Thu et al., 1997). Recent air quality studies have shown that CAFOs can adversely affect air quality through the release of odor (Jacobson et al., 1997b; Zahn et al., 2001) and odorous compounds such as hydrogen sulfide (H_2S) (Jacobson et al., 1997a), ammonia (NH_3) (Asman, 1995; Eklund and LaCosse, 1995; Sharpe and Harper, 1998), and volatile organic compounds (VOCs) (Zahn et al., 1997; Zahn et al., 2001).

Efforts to remediate odor from swine production facilities have been impeded by the lack of instruments capable of high-throughput, objective odor measurements. The desire to develop high-throughput, inexpensive methods of odor quantification has been the impetus for several recent investigations that have focused on defining relationships between gas concentration of odorants emitted from animal manure and odor intensity measured by olfactory methods (Hobbs et al., 1995; Jacobson et al., 1997a,b; Obrock-Hegel, 1997; Pain et al., 1990; Zahn et al., 2001). Obrock-Hegel (1997) found that nutritional manipulation of amino acid intake reduced NH_3, cresols, and indoles measured in air samples from production environments. However, no reduction in odor concentration was observed between control and treatment samples. Schulte et al. (1985) and Hobbs et al. (1995) linked high levels of NH_3 to odor. Unfortunately, the latter authors noted that the relationship

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Abbreviations: VOC, volatile organic compound; CAFO, concentrated animal feeding operation.